Ground Water Monitoring&Remediation

Testing the In-Well Horizontal Laminar Flow Assumption with a Sand-Tank Well Model

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Abstract

The assumption of horizontal laminar flow within a monitoring well is a commonly cited basis for interval sampling using low-flow or no-purge sampling techniques. A few studies have shown horizontal flow over short distances within the well for short periods of time. Others have demonstrated specific circumstances under which the assumption fails. But surprisingly, little focus has been given to confirming the underlying concept—that under "normal" conditions (i.e., no vertical hydraulic gradient) water enters one side of a well and exits the other side of the well at the same elevation. To test the horizontal flow assumption, a physical sand-tank model was constructed to observe flow through in a simulated monitoring well. The well, filter pack, and aquifer largely mimic real world conditions of a submerged well in a moderately high-permeability sand. To observe flow behavior in the simulated well, a dye "stringer" was introduced into an injection port upgradient of the simulated well. In all tests, regardless of flow rate or small density differences, the dye stringer eventually mixed throughout the model monitoring well. Since the model approximates a section of an at-scale well subjected to real world bulk flow rates, mixing appears to be the rule rather than the exception for near–neutrally buoyant contaminant stringers in homogeneous flow fields. Despite additional heterogeneities introduced by field conditions, there are several clear and important implications of this study: (1) some degree of in-well mixing and flow-weighted concentration averaging may occur in a well before any purge or sampling efforts are made; (2) in-well mixing may mask low to moderate contaminant stratification in an aquifer; (3) contaminant stratification, if present inside a well, implies strong contaminant stratification outside the well; (4) contaminant stratification inside a well may not correspond to stratification at the same interval outside the well; and (5) vertical stratification within an aquifer may not be accurately monitored by sampling multiple intervals within an open well.

Introduction

Although new thoughts are developing currently, low-flow and passive sampling techniques are conventionally thought to represent contaminant concentrations at intervals of the aquifer near the pump intake or sampler deployment position. The conventional thought for the low-flow and passive sampling largely assumes horizontal laminar flow within a monitoring well—that water from the aquifer enters the well, flows horizontally, and exits the well at roughly the same elevation (e.g., see introduction of Parker and Clark 2002, 2004).

The study presented here tests aspects of this critical assumption. How does a contaminant "stringer" behave when it enters a ground water monitoring well in natural background flow conditions? Does it flow straight across the well? Or does it mix and dilute with cleaner water from other intervals? In this study, mixing and dilution are visually observed using time-lapse photography of a dye tracer as it enters a model monitoring well. Limited quantitation of dye fluorescence was performed to ground the visual observations.

Background

Purging monitoring wells to collect representative ground water samples has been a conventional procedure for more than a generation (e.g., U.S. EPA 1977; USGS 1980). Over much of that period, some level of controversy has continued over how best to purge wells or whether it is necessary to purge wells at all (Robin and Gillham 1987; Powell and Puls 1993; Newell et al. 2000). Many studies have identified how anomalous or otherwise unrepresentative results may be generated from traditional purge-and-sample techniques (e.g., Robbins 1989; Reilly et al. 1989; Gibs and Imbrigiotta 1990; Reilly and Gibs 1993; Gibs et al. 1993; Conant et al. 1995; Church and Granato 1996; Martin-Hayden and Robbins 1997; Reilly and LeBlanc 1998; Hutchins and Acree 2000; Elci et al. 2001, 2003).

Common situations shown by these and other studies have caused investigators to question what traditional purge-and-sample ground water monitoring results represent. Low-flow purge techniques (e.g., Puls and Barcelona 1996) were developed and widely adopted to address some of the problems, with the added benefit of reducing purge-water waste. No-purge techniques are also being explored and are being adopted where applicable (Vroblesky 2001; Parker and Clark 2002, 2004; Parsons Engineering Science 2003; Interstate Technology and Regulatory Council 2004). These alternative techniques can solve some problems like elevated turbidity and volatile organic compound loss caused by bailer agitation or high pump rates, but they do not solve problems with vertical flow (Elci et al. 2001, 2003) or pumping-induced variability (Martin-Hayden 2000a, 2000b; Gibs et al. 2000). Other techniques involve installing multichannel tubing wells (Einarson and Cherry 2002), short-screen direct-push wells (Kram et al. 2001), or devices such as the discrete multilevel sampler (DMLS) within existing longer screen wells (Puls and Paul 1997). These techniques are effective but may require a long-term commitment to relatively expensive multiinterval sampling.

Despite extensive study on purge sampling, little focus has been given to passive mixing within the open-well casing of monitoring wells between sampling events. Mixing within open-well casings is of concern for two main reasons: redistribution of contaminants and dilution of high-concentration contaminant stringers with surrounding cleaner water. The first concern, redistribution of contaminants, is inherently obvious but particularly important for longer screen monitoring wells. The second concern, dilution, is project specific, depending on data objectives. The effect and ramification of dilution is especially important for objectives such as sentinel well detection monitoring.

The Working Hypothesis for This Study

The working hypothesis for this study was that under simple scenarios, contaminant mixing and dilution do occur in monitoring wells under natural flow-through conditions.

The flow-through aspect of this hypothesis is more generally accepted, as American Society of Testing Materials (ASTM) observes in its 2002 standard practice document for low-flow purging:

> Low-flow purging ... is based on the observations of many researchers that water moving through the formation also moves through the well screen. Thus, the water in the well screen is representative of the formation surrounding the screen (ASTM 2002).

The mixing/dilution aspect is not as well known—and was the reason for the study. A number of well stratification studies have been conducted, which yielded evidence of contaminant stratification in some wells but not in others (e.g., Parsons Engineering Science 2003; Vroblesky and Peters 2000). Little direct information was available in these studies, however, regarding corresponding aquifer contaminant stratification. A study conducted by Puls and Paul (1997) included sampling both well samples collected with in-well barriers (i.e., DMLS) and discrete interval samples outside the well (direct-push well sampling). This study illustrated corresponding stratification, but the DMLS system prevented mixing within the wellbore itself. Little focus has been given to aquifer stratification vs. open-well stratification. This study tests the mixing/dilution hypothesis.

Sand-Tank Model Construction and Operation

The simple scenario selected to test the in-well mixing hypothesis was a fully saturated short-screen well, set in a substantially homogeneous sand. Homogeneity of inflow velocity into the well was the key objective for this study. While many potential conditions exist in true field conditions, homogeneity was selected as a baseline to test the mixing effects under these simple conditions. Common bulk flow velocities were chosen to simulate real world conditions. Consideration of strong lithologic heterogeneity (and resulting heterogeneous inflow velocity) was deferred to another study. Other differences between the model and "real world" scenarios are discussed later in this paper. The well model for this study was constructed to meet the homogeneous inflow criterion, as described in the following sections.

Sand-Tank Model

The sand-tank well model was constructed using commercially available raw materials including lumber and plywood for the support structure; polycarbonate sheet stock (Lexan), angle polyvinyl chloride (PVC) structural materials for the tank; PVC and polyethylene fittings and valves; Tygon® tubing, aquarium glue, 0.025-cm slot Johnson Screens' PVC VeeWire well screen, and two types of filter sand.

Figure 1 is a photograph of the modeled aquifer and well. The model was designed to simulate a vertical cross section through a 10-cm monitoring well. Water flow could be induced in the model aquifer and well by controlling head and/or flow in the upgradient and downgradient reservoirs or in the well. The overall dimensions of the system include two 50-cm-wide by 72-cm-high by 7-cm-thick "aquifers" on either side of a 10-cm well. The reservoirs and the well are screened across the entire 7-cm by 72-cm cross section of the aquifer with 0.025-cm slot screen. The aquifer sand is a well-sorted medium sand. Coarser filter pack sand was placed adjacent to the well screens to minimize movement of the aquifer sand into the well and to more closely simulate a field installation of a filter-packed well. The filter pack was 5 cm wide adjacent to both reservoirs and adjacent to both upgradient and downgradient sides of the well.

The polycarbonate sheet and PVC angle material were cut and sized to function as the support for the aquifer sand and well. Internal spacers were bolted in place between the front and rear walls of the model to maintain model width and for structural strength. The well screen was keyed in place with slots in the front and rear walls of the model. The model was attached to the wood support structure using galvanized or stainless steel hardware to

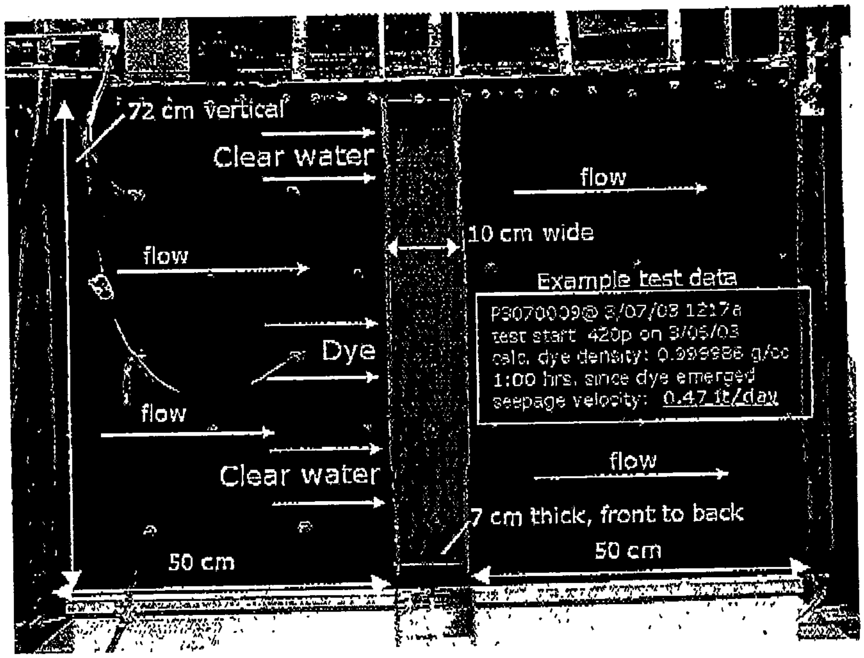

Figure 1. Set-up and dimensions of sand tank well model.

avoid corrosion. Rubber plumbing gasket material was used with the hardware to avoid leaks. Six dye injection ports were installed in the upgradient aquifer. The dye ports incorporated diffusers that encourage dye introduced to the model to flow through the width of the model before moving downgradient in the sand. All joints in the plastic were sealed with aquarium glue. Watertight seal was tested before adding the aquifer sand. Aquifer and filter pack sand was added to the model by gravity feed from the top of the model. The model was agitated as the sand was added to settle the sand. Care was taken to maintain horizontal homogeneity as much as feasible using gravity-feed placement. The method of settlement may have induced some vertical/horizontal heterogeneity, but this effect was not considered critical because vertical flow was anticipated to be minimal in this study. Testing results showed little or no vertical flow in the model, verifying the initial assumption.

Once the aquifer and filter pack sand was installed and settled, a top cover plate was sealed in place. Piezometers were placed along the top plate to allow measurement of head in the top of the aquifer and the well. Head in the upgradient and downgradient reservoirs was observable directly. Head in the middle of the aquifer could be measured from the dye injection ports.

The model was filled with tap water slowly from the bottom to limit entrained air in the aquifer sand. Water level throughout the model was maintained above the top plate of the model to simulate a confined system. The submerged configuration was preferred to eliminate flow differentials that may have occurred because of a capillary fringe. The model well itself was closed at the top except for a 1.25-cm piezometer. Therefore, there was no saturated blank casing above the screen interval in the well.

Flow Control

Flow through the model was controlled by inducing a gradient between the upgradient and downgradient reservoirs. The upgradient reservoir head was maintained at a constant level by connecting it with a tubing to a separate constant-head reservoir. The water level in the reservoir was maintained at a constant level by means of a drain set at a fixed elevation and a circulation pump continually returning drained water back to the reservoir.

Head in the downgradient reservoir was controlled by draining the reservoir at variable rates. The discharge rate from the downgradient reservoir was metered using a needle valve and a 2-L graduated cylinder. The decline in head in the downgradient reservoir induced flow through the model and resulted in a fixed gradient across the model within several minutes of the start of discharge.

Discharge rates were calculated by periodically measuring total discharge volume and time since last measurement. Seepage velocities within the simulated aquifer were calculated using an assumed effective porosity of 0.25 in the aquifer. Thus, the seepage velocity for each test run was determined by dividing the porous cross-sectional area ($504 \text{ cm}^2 \times 0.25 = 126 \text{ cm}^2$) by the discharge rate (cm^3/d). Most of the test runs were conducted at seepage velocities ranging from ~10 to 40 cm/d (35 to 150 m/year). Visual observation of dye progression in the flow field reasonably supported these calculations.

Dye Mixing and Density Control

The contaminant proxy chosen for these tests was Rhodamine WT dye. The dye is fluorescent and visually observable at very low concentrations (100 ppb). There is a linear relationship between the dye concentration and its fluorescence. It has strong color saturation at higher concentrations and is therefore effective for use with color photography.

The raw dye selected for these experiments was a 5% Rhodamine WT dye solution. The manufacturer reports a density of 1.0375 g/cc at 20°C. For these tests, the raw dye was mixed with tap water at a ratio of 1.0 mL raw dye to 3500 mL water. The density of the tap water was not measured. Rather, the density of the tap water was arbitrarily set at 1.000000 g/cc. This was acceptable because the difference in calculated density induced by adding the dye to water with a density of 1.000500 g/cc vs. water with a density of 1.000000 g/cc is negligible. To maintain density consistency, a reserve of tap water was collected for each test run. The same tap water was used for mixing the dye and for the clear water used in each test run.

The density of the dyed water was always higher than the clear water unless the dyed water density was reduced. Density reduction was accomplished by adding small amounts of methanol to the dye to create a neutrally buoyant dye or a "light" dye. Calculated dye densities used in these experiments ranged from 0.999935 to 1.000011 g/cc. For reasons that are unclear, but likely because of density error in the raw dye, purity of the methanol, or other factors, density calculated to be around 0.999975 g/cc behaved neutrally buoyant. Densities lower than this benchmark behaved lighter than the surrounding clear water and densities higher behaved heavier.

Density is also affected by very small changes in temperature. For this reason the tap water, dye, and tank were always allowed to come to equilibrium temperature for 1 to 2 days after drawing new tap water for the tests. Additionally, dye added to the model was allowed to temperature equilibrate in the aquifer for 12 to 24 hours as it moved toward the well. Observations of the behavior of the dye as it entered the well, lack of differential flow in the front or back of the model, and lack of development of any features resembling a convection cell indicated that temperature-induced variability was not a significant source of experimental error in the model.

Digital Time-Lapse Photography

Using Pine Tree Computing's camera controller shareware, time-lapse photographs were taken of the model during each test run. Photographs were taken at intervals ranging from 5 minutes to 1 hour, depending on the flow rate of the test and other factors. The time-lapse photography recorded the dye trace as it progressed horizontally from the injection port to the well and then the behavior of the dye in the well. In all cases, the dye mixed and diluted to a degree that it was not visually observable in the aquifer downgradient of the simulated well.

The photographs served several purposes: they were digitally time-stamped so that time progression was recorded, photographs could be viewed in rapid succession to see behavior not observable in real time, photographs would retain a record of the dye behavior under different flow rates and dye densities, photographs would facilitate comparison of different behaviors, and they could be used to test repeatability.

How the Model Differs from a Real World Scenario

The model is intended to simulate a 10-cm well installed below the water table in a moderately high-permeability sand. The model well differs from a three-dimensional field installation of a well. This was necessary so that fluid behavior could be visually observed in cross section. This was accomplished by constructing a 7-cm-thick vertical cross section of a sand aquifer through the center of a monitoring well. There is no simulated aquifer in front or behind the well. Horizontal flow is therefore forced to go through the model well. Field conditions would permit water to preferentially flow around or through the well as dictated by the conditions and well construction for the individual well.

Flow forced through the model well is both an advantage and disadvantage for this testing regime. The obvious advantage is that fluid behavior is observable through the front cover plate of the model. The disadvantage is that flow into and out of the well is closer to a two-dimensional flow field rather than a three-dimensional system. A three-dimensional system would allow lateral distortions in the flow field due to differing permeabilities in the sand pack and well itself (Graw et al. 2000). These lateral distortions are not represented in this model. This disadvantage is not a critical flaw because it also limits artifacts such as borehole skin effects and angled flow vectors. An additional effect of forcing flow through the well is that bulk velocity in the well is slower than in the aquifer. This is the opposite of what is expected in a three-dimensional flow field—where nonsquare flow vectors and higher differential flow rates promote mixing (Graw et al. 2000). In this respect, the model well is more conservative than field conditions because inflow vectors were essentially square and flow slows as water enters the well.

The model well is 72 cm long (high), with the screen section completely submerged and no saturated blank casing. This size is about half the length of common short-screen wells. Many wells are four or even eight times longer. This was considered acceptable because the size of the model well is within a factor of two or four of common monitoring wells. Fluid behavior is anticipated to be similar within this narrow size range. The lack of saturated blank casing is an artifact by definition for a submerged screen well. This was considered acceptable in order to isolate the observed phenomena. If mixing within saturated blank casing was to be considered, an additional variable would need to be reconciled. This was not considered necessary for this study. Other investigations have considered the problems of interaction of "live" screen water and "stagnant" saturated blank casing water (e.g., Martin-Hayden 2000b).

Several real world variables are evident but were not tested. These include permeability heterogeneity, screen vs. blank sections in the well, a water table with capillary fringe, vertical percolation of dye contaminant, etc. These were not tested in this study. The intent of this study was not to test a large variety of real world possibilities but rather to limit variables so that clearer conclusions could be drawn from the specific variables tested.

Model Runs

Three basic scenarios were chosen for test runs with the sand-tank well model. Neutrally buoyant dye, slightly lower density dye, and slightly higher density dye were each added to the aquifer upgradient of the model well to assess the dye behavior as it entered and moved across the well under horizontal ambient flow. Individual runs for each density dye were conducted by injecting dye in the model aquifer upgradient of the vertical center of the well screen.

Neutrally Buoyant Test Runs

Neutrally buoyant dye runs used a dye mixture that did not display an initial unidirectional vertical displacement. There was vertical movement of the neutrally buoyant dye stringer as it entered the well, but the direction of movement was not uniformly up or down. Figures 2A and 2B show the progression of a neutrally buoyant dye trace over ~4.5 days. Seepage velocities for this test ranged from 9 to 31 cm/day. Most of the test, except for the last 20 of the 106 hours, was run at a fairly consistent flow rate between 12 and 17 cm/day.

In each case where testing was conducted using neutrally buoyant dye, the dye entered the well very slowly at the vertical interval corresponding to where the dye was introduced in the model aquifer. Very little vertical dispersion occurred in the model aquifer or filter pack before the dye moved into the model well. The dye progressed through the simulated aquifer without acceleration or retardation at the model walls—there appeared to be no boundary ("skin") effects. Once the dye began to enter the model well, it spread vertically in both directions at a much higher rate than its progression across the well. The neutrally buoyant dye moved from the central entry point to both the top and bottom of the model well in 4 to 10 hours—a distance of 20 to 30 cm. The dye progression across the entire length of the well was much slower, with horizontal displacement of 10 cm in 20 to 40 hours. This horizontal velocity is consistent with the overall horizontal flow velocity for the model aquifer.

The bulk of the results for this study was qualitative, using visual color saturation as a gauge of stratification and mixing. For one test run, however, results were quantified using Rhodamine WT fluorescence. Availability of

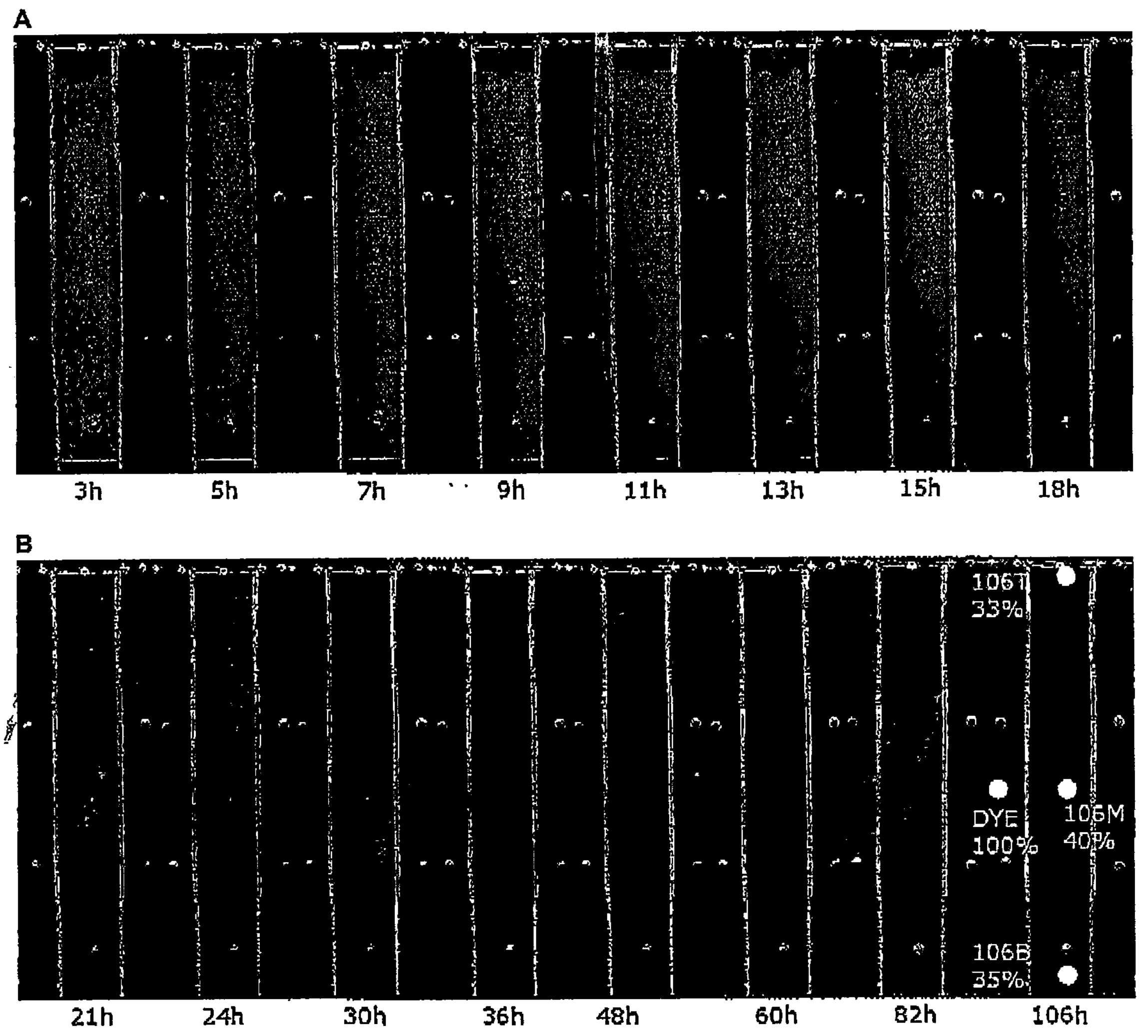

Figure 2. (A) Neutral buoyancy test run, first 18 hours. (B) Neutral buoyancy test run, continued (21 to 106 hours). Photograph captions indicate the number of hours since dye emerged in the model well. The data points identified as DYE, 106T, 106M, and 106B are locations where fluorescence samples were collected at the end of this test run. Percentages indicate dye saturation relative to the injected dye concentration.

the fluorometer was limited, so only one set of fluorescence data was collected at the end of the neutrally buoyant test shown in Figure 2B. A sample of the initial dye and samples from the top, middle, and bottom of the model well were collected at the end of this neutral density test. The concentration in each interval of the well was compared to the full-strength initial dye concentration to gauge dilution and stratification.

The last photo in Figure 2B identifies the collection points for four samples taken at the end of this test. The initial dye stringer thickness as it entered the well was ~25 cm (35% of the aquifer thickness). Resulting dye concentrations in the well showed the following: 33% of the initial dye concentration at the top of the well, a concentration of 40% of the initial dye concentration in the middle of the well, and a concentration of 35% of the initial dye concentration at bottom of the well.

These results indicate that while a slightly higher concentration is present at the interval where the dye enters the well, the difference is minor compared to the overall dilution and mixing throughout the well during the test. For the test quantified with fluorescence results, the dye-saturated thickness in the model aquifer corresponds to the degree of dilution in the well. This correspondence indicates proportionate mixing of dye and clear water—a flow-weighted averaging effect. Similar mixing and dilution results were consistently observed in those tests not quantified with fluorescence testing.

Differential-Density Test Runs

Low density tests were run with injected dye slightly below neutral buoyancy. Test runs ranged from ~1 to 4 $\times$ 10^{-5} g/cc less dense than neutral. A density difference of this magnitude was sufficient to cause an initial upward vertical displacement of the dye as it entered the model well. Similarly, high-density dye stringer tests were conducted with dye density slightly higher than the surrounding water. Test runs ranged from ~1 to 4 $\times$ 10^{-5} g/cc more dense than neutral. A density difference of this magnitude was sufficient to cause an initial downward vertical displacement of the dye as it entered the well screen. Figures 3 and 4 show the progression of the dye in each of the differential-density scenarios.

In each case where testing was conducted using low- and high-density dye, the dye entered the well very slowly at the vertical interval where the dye was introduced in the model aquifer. Like the neutrally buoyant dye, very little vertical dispersion occurred in the model aquifer before the dye moved into the model well. Once the dye began to enter the model well, the dye trace moved in the direction that was predicted based on its relative density. For the low and high relative density dyes, the dye moved through the whole length of the well in 10 to 20 hours—a distance of 20 to 30 cm. The dye progression across the entire length of the well was much slower with horizontal displacement of 10 cm in 20 to 40 hours. This velocity is consistent with the overall horizontal flow velocity for the model as a whole and is consistent with velocities tested in the neutral-buoyancy tests.

Key Observations

- Dye stringers do not maintain their integrity as they pass into and out of the model well at all tested flow rates and dye densities—"horizontal laminar flow" was not observed within the model well, despite horizontal flow in the simulated aquifer.
- Dye stringers introduced into the sand-tank well model mixed throughout the model well in 24 to 96 hours.
- Once mixed, the well remained so for several days until each test was terminated. No tests showed restratification once the water column mixed.
- Some degree of mixing throughout the model well was observed in all tests, despite initial stratification caused by small density differentials in the test dye.
- Qualitative results of in-well mixing show a flow-weighed averaging effect of the dye stringer and the clear water concurrently entering the model well.

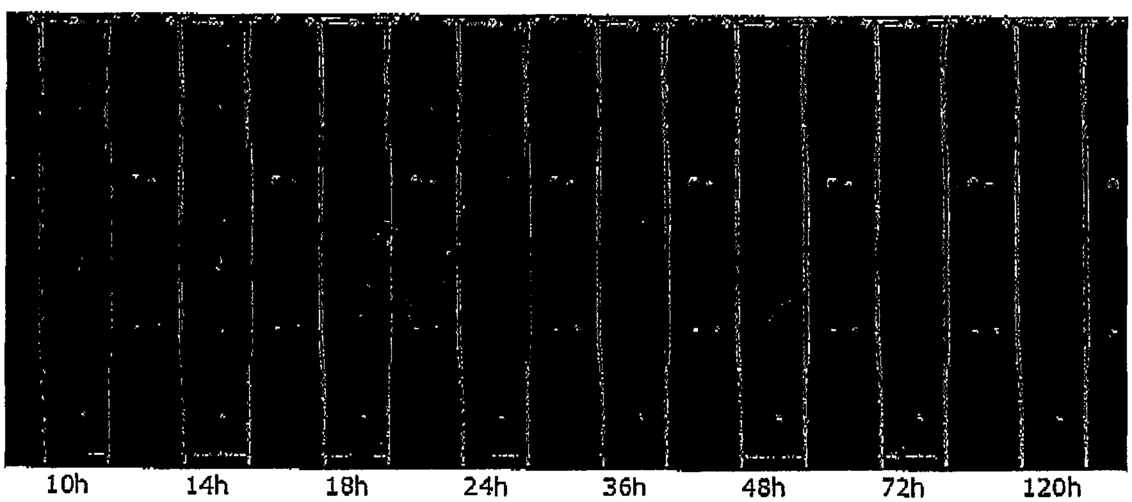

Figure 3. Low-density test run. The photograph captions indicate the number of hours since dye emerged in the model well.

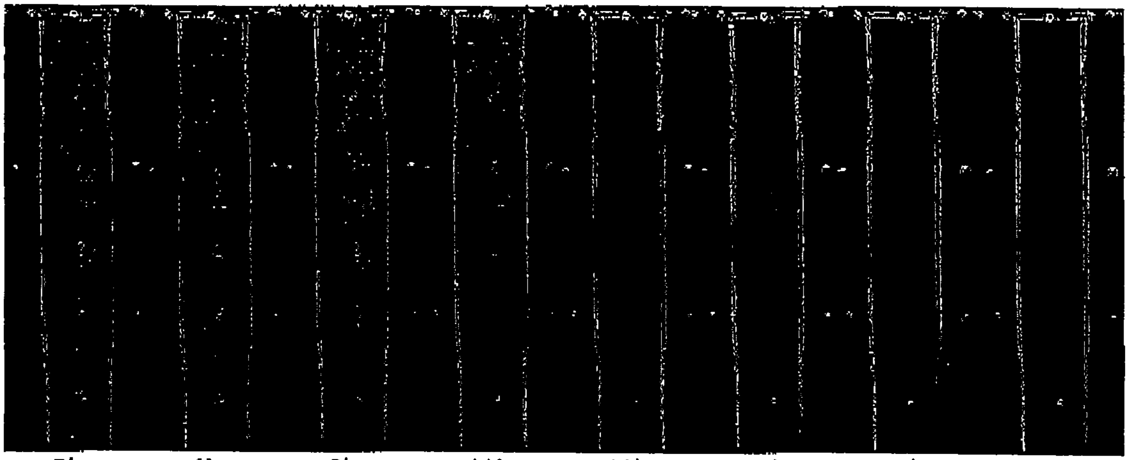

Figure 4. High-density test run. The photograph captions indicate the number of hours since dye emerged in the model well.

- Limited quantitative fluorescence testing shows flow-weighted concentration averaging and dilution of the dye stringer, supporting the qualitative results.
- Dye concentration in the well at the point the dye entered the well was slightly higher than other portions of the well, but mixing and dilution were clearly dominant.

Discussion

It is difficult to convey the very slow speed of the in-well mixing process. The movement of the dye is imperceptible in real time. This observation is notable because it is difficult to imagine when looking at time-series photographs of the entire progression. It is important because conventional concepts of turbulent flow or mixing caused by water "shooting" through the well screen are not effective explanations for the observed phenomena. These effects may be occurring, but if they are, they occur on a scale that is not visually observable.

Advection appeared to be the dominant transport mechanism in the studies. There was no perceptible vertical dispersion or diffusion of the dye slug before it entered the well. Diffusive softening was observable as a halo around the dye in the well, but diffusion was not the driver of mixing, otherwise the color progression would have moved across the well as fast as it moved vertically. No apparent convection was observed in the model. "Skin effects" from the casing or model walls either propelling or slowing the dye were not observed.

The density differences tested here approached zero and were virtually immeasurable. The aforementioned densities were in fact calculated based on solute contribution rather than directly measured. The density differences are small enough that one would expect these differentials to be quite common over a meter or less in an aquifer in the natural environment. The point where density differences begin to cause continued stratification was not tested in this study, but ongoing work focuses on heterogeneities, including inflow velocity heterogeneity, that may cause contaminant stratification within wells.

Implications

- In-well mixing and dilution in an open well may mask contaminant stratification in an aquifer.
- Due to in-well mixing, aquifer contaminant stratification may be more common than in-well stratification testing implies.
- Under unpumped background flow conditions, stratified contaminants within an aquifer show a tendency to mix within a monitoring well proportionate to their flow-weighted contribution.
- Contaminant stratification, if present inside a well, implies strong contaminant stratification outside the well.
- Contaminant stratification measured inside a well may not correspond to stratification at the same interval outside the well.
- Vertical stratification within an aquifer may not be accurately monitored by sampling multiple intervals within an open well.
- In-well stratification, where present, is likely a result of much higher density differences than tested here or a result of factors such as flow rate heterogeneity, temperature stratification, off-gassing within the well casing, vertical flow, or other causes.
- Barriers to in-well mixing may limit mixing and assist in defining intervals of aquifer contaminant stratification.

Conclusions

This study provides some insight into behavior of ground water in the relatively brief period it resides in a monitoring well. Mixing and dilution were observed. Horizontal laminar flow was not observed. Additional work is needed to verify these effects in the field, and questions remain unanswered regarding the detailed causes of stratification within monitoring wells. Nonetheless, the limited results presented here are compelling in that they demonstrate in a visually effective way that water in an open water column mixes when subjected to real world aquifer flow rates and density contrasts.

Mixed water columns within wells potentially affect sampling data interpretation in the following ways. Passive samples may reflect zones of the aquifer beyond the interval where the sampler is deployed. Pump discharge from the screen interval, either by high flow or low flow, may start at a flow-weighted average early in the purge cycle, when most water in the screen interval is purged by ambient flow through; depart from that flow-weighted average as stratified formation water mixes with unstratified well water and then become flow-weighted again when purging reaches stability.

If ambient flow through and mixing creates a flow-weighted average concentration in the screen interval of a well, it is arguable that wells are "naturally purged." Simple tracer dissipation tests could determine whether individual wells have natural flow through. Passive stratification testing can show whether contaminant concentrations in the well are mixed. Naturally purged wells could allow collection of the ideal sample—a single, inexpensive representative sample collected directly from the screen interval of the well.

Naturally purged wells with true flow-weighted mixing conditions may prove to be rare, but additional investigation is needed to evaluate how rare it is. With the horizontal laminar flow assumption, those ideal conditions would have been impossible with any level of aquifer contaminant stratification. If well water in a physical model can mix to create a nominally flow-weighted average concentration, perhaps it can be shown that those ideal conditions are not quite so rare in field conditions.

It is not within the scope of this paper to elaborate on a variety of scenarios, such as the effect of pump placement or purging, the effect of blank casing water, or the effect of well headspace on well water chemistry. But for effective use of well sampling data, it is clear that a fair amount of thought should go into a well's construction, local lithologic variability, sampling procedure, and purge parameter trends when interpreting what a well's contaminant data represent. It is attention to these factors that is likely to yield better decision making on important topics such as sampling method, pump or passive sampler placement, and ultimately, remedy selection.

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Disclaimer

This paper reflects the opinions of the author. The Department of Toxic Substances Control does not take responsibility for the opinions of the author, nor does it necessarily accept or concur with the findings.

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